Evaluating whether there are trade-offs between plant diversity and ecosystem functions in restored and unrestored Lake Erie coastal wetlands

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Abstract

Coastal wetlands along Lake Erie have been dramatically altered by humans, disrupting important natural ecosystem functions including habitat provision, flood mitigation, and nutrient retention. Restoration actions, such as the removal of dikes, aim to restore these natural processes. While the goal of dike removal is to restore long-term ecosystem functioning, there may be short-term trade-offs between restoring particular ecosystem functions and maintaining biodiversity. Higher-than-optimal water levels and longer inundation periods following hydrological reconnection may increase nutrient retention, but decrease wetland plant diversity. This could affect primary productivity and nutrient uptake by wetland plant communities, thus affecting higher trophic levels in the wetland ecosystem and water quality in Lake Erie (e.g. higher levels of nitrogen and phosphorus). The goal of this study was to compare wetland inundation, nutrient levels, primary productivity, and plant diversity in restored and unrestored coastal wetlands over the course of a growing season. We hypothesized that the restored wetlands would have higher water levels and higher levels of nitrogen and phosphorus, but lower primary productivity, and lower plant diversity than the unrestored wetlands. Twelve coastal marsh sites – 5 restored and 7 unrestored – were sampled in the Ottawa National Wildlife Refuge wetland complex in Oak Harbor, Ohio. We measured water level and collected water samples biweekly between May-August, 2017. Water samples were analyzed for total nitrogen and phosphorus concentrations. Plant diversity surveys were conducted in June and July, and peak, above-ground biomass of emergent plants was measured in mid-August. Results suggest that mean water depth and its relative changes did not differ between restoration status, and that total nitrogen and levels were lower in restored sites than in unrestored sites. Wetland plant taxonomic richness was lower in restored wetlands, but overall diversity and peak biomass was similar between restored and unrestored wetlands. Overall, significant short-term tradeoffs were not identified as a result of hydrological reconnection, but long-term monitoring of the wetlands will need to occur to evaluate potential long-term tradeoffs in wetland plant community diversity and functioning.

Introduction

Historically, coastal wetlands of Lake Erie supported important ecosystem functions, such as flood mitigation, water quality improvement, and maintenance of biodiversity (Zedler & Kercher, 2005). After the mid-1900s, inland wetlands were largely converted for agricultural purposes. The remaining coastal wetlands were diked to protect farmlands from flooding, resulting in further degradation and loss of ecosystem functions performed by wetlands. Concern over the increasing frequency and intensity of harmful algal blooms (HABs) in Lake Erie has led to recent efforts in the Western Lake Erie Basin (WLEB) to reconnect coastal wetlands in the Ottawa National Wildlife Refuge (ONWR) to the Crane Creek watershed and Lake Erie, thus restoring important wetland functions (e.g. nutrient retention in wetlands). However, there is evidence that restoration of ecosystem functions may lead to short-term trade-offs in biodiversity. For example, removing dikes from wetlands has been shown to cause mass vegetation die-offs due to higher-than-optimal water levels that impair seed germination and plant growth (Sherman et al, 1996).

Understanding the relationship between biodiversity and ecosystem functions has received much attention due to growing concerns that loss of biodiversity may impair ecosystem functioning (Yachi & Loreau, 1999). It is well understood that biodiversity is one of many factors that drive ecosystem functioning (Tilman, 1999). Recent conceptual advances in biodiversity-ecosystem function (B-EF) research provide a powerful framework that can be applied to identify short-term trade-offs that follow restoration efforts (Woodward, 2009). For example, more diverse wetland plant communities have been shown to retain higher amounts of nutrients (i.e., reduce nutrient run-off into adjacent rivers and lakes) (Barry et al, 2004; Mitsch et al, 2005). Yet, short-term changes and increases in water levels following hydrological restoration of wetlands (e.g., dike removal) have been shown to decrease plant biomass and diversity (Sherman et al, 1996). Therefore, following the B-EF framework, hydrological reconnection of coastal wetlands may increase water levels within wetland pools and alter wetland plant diversity and biomass, relative to unrestored coastal wetlands. Furthermore, short-term reductions in wetland plant diversity and biomass may be associated with reductions in ecosystem functions (i.e. nutrient retention).

Anthropogenic land use change has blocked the progression of wetlands inland, which is a mechanism necessary for wetlands to adapt to variations in Lake Erie water levels. Historically, long-term and short-term water level fluctuations drove the maintenance of wetland plant biodiversity because periodic high lake levels eliminated competitively dominant species, allowing less competitive species to repopulate (Bloczynski, et al 2000). Now, high water levels can result in the net loss of wetland habitat rather than redistribution. It has been suggested that many coastal wetlands in western Lake Erie would have been destroyed had they not been diked (Herdendorf, 1987). Water levels in diked wetlands are controlled to simulate natural fluctuations that promote seed germination and plant growth critical for migratory birds and waterfowl. Hydrologic reconnection may result in higher-than-optimal water levels that inhibit germination and growth of wetland plants. Wetland plants are responsible for most ecological functions found in wetlands including nutrient cycling, food and habitat for invertebrates and waterfowl, and reducing erosion (Herdendorf, 1987). Long-term flooding resulting from hydrologic reconnection may reduce the abundance of dominant, monocultural plants; however, without periods of low water, other species do not have an opportunity to colonize from the seedbank present in the sediment. In a study conducted at the ONWR, it was found that very little emergent plant diversity was present in times of continual flooding, and the species richness found in the seedbank was reduced (Barry et al 2004).

In hydrologically connected wetland sites in the Crane Creek watershed, it was found that average nitrogen concentrations can fluctuate between 0.02 – 3.19 mg/L, with lower levels associated with temporarily elevated water levels, such as those caused by flood events (Kowalski et al, 2014). Wetland plants receive most nitrogen and phosphorus necessary for growth from sediment rather than directly from open water, because nutrients must first be adsorbed by sediment (Johnston 1991; Kowalski et al 2014). While wetland plants help mitigate excess nutrient loads, regular influxes of nutrients cannot be fully mitigated by vegetation. A decline in wetland plant diversity in restored wetlands due to irregular inundation and high water levels could result in fewer nutrients being retained by the system.

The main objective of this study is to evaluate potential short-term trade-offs between restoring ecosystem functions (i.e. nutrient retention via hydrological reconnection) and the structure and function of wetland plant communities in coastal wetlands. This is done by comparing wetland inundation, plant species richness and diversity, plant biomass, and nutrient concentrations between restored and unrestored coastal wetlands over the course of a growing season.

Objective 1: Quantify differences in water levels between restored and unrestored coastal wetlands in the ONWR.

Hypothesis 1: Restored wetlands will have higher water levels than unrestored wetlands.

Objective 2: Quantify differences in wetland plant species richness, diversity, and biomass between restored and unrestored coastal wetlands in the Ottawa National Wildlife Refuge.

Hypothesis 2: Wetland plant species richness and diversity will be lower in restored wetlands in comparison to unrestored wetlands following hydrological restoration.

Hypothesis 3: Restored wetlands will have lower wetland plant peak biomass than unrestored wetlands.

Objective 3: Evaluate the relationship between wetland inundation, plant species richness and diversity, peak biomass, and nutrient concentrations across restored and unrestored wetlands.

Hypothesis 4: Wetlands with more inundation will have lower wetland plant diversity and peak biomass and thus, greater nutrient levels (i.e. higher total nitrogen and total phosphorous concentrations).

Methods

Study Site

The Crane Creek watershed in north-central Ohio (Wood County) is approximately 146 km^2 and drains into Lake Erie. At the delta of Crane Creek, about 345-ha of coastal wetlands are located within the U.S. Fish and Wildlife Service Ottawa National Wildlife Refuge (ONWR) along the southern shore of the western basin of Lake Erie. The primary land-use within the Crane Creek watershed is agricultural and the southern extent of the coastal wetlands is bordered by farmland.

Field sampling was conducted within twelve wetland sites in the Crane Creek wetland complex located in the ONWR (Figure 1). Seven of the sites continue are hydrologically disconnected from the watershed with the use of dikes, and their water levels are regularly managed by pumping water in and out as necessary. Five of the sites have been hydrologically reconnected from 2011 to 2017. In each of the wetlands, water levels, nutrient levels, wetland plant species richness and diversity, and wetland plant biomass were sampled to compare restored and unrestored wetlands.

Figure 1: *ONWR coastal wetland site map. Unrestored wetlands are labeled in orange and restored wetlands are labeled in blue. White arrows depict the Crane Creek connection to Lake Erie.*

In addition to sampling the restored and unrestored wetland pools, water levels and nutrient concentrations were also sampled in the mainstem channel of Crane Creek, which is adjacent to the wetlands of interest.

Water Levels and Nutrient Concentrations

Water levels and nutrient concentrations were measured on a biweekly basis from May to August. For each wetland site, an easily accessible open-water area was identified from the shore. A permanent transect was then established perpendicular to the shore, beginning at the water's edge and extending 15m into the open-water area. Water depth (cm) was measured at five regular intervals along the transect.

At the end of the 15m transect, a water grab sample was collected, which was immediately placed on ice and then frozen. All water samples were sent to OSU's Service Testing and Research (STAR) Lab for measurement of total phosphorous and total nitrogen concentrations (mg/L). Additional water quality parameters measured were temperature (°C), conductivity (microS), dissolved oxygen (mg/L), pH, and turbidity (NTUs).

Plant Species Richness and Diversity

Plant species richness and diversity was sampled in mid- to late-June to facilitate better species identification with plant reproductive parts (Little, 2013). For a more accurate representation of the overall plant species richness and diversity of restored and unrestored wetlands, a stratified sampling approach was taken in which each of the three major marsh vegetation types (zones) – submergent, floating, and emergent – were sampled in all twelve wetland sites (Stohlgren, 2006; Lougheed et al., 2001). This was done by visually assessing each wetland from the shore and subsequently choosing three sampling locations with distinct submergent, floating, and emergent plant zones, respectively, that were representative of the wetland.

Within each vegetation zone, three square-shaped, 0.5 m^2 quadrats were used to collect plant species richness and diversity information; therefore, richness and diversity was sampled in nine quadrats in each of the twelve wetland sites. Quadrats were placed at least one meter from the vegetation zone boundaries, and they were placed two meters apart from other quadrats to avoid trampling vegetation that was to be sampled (Uzarski et al., 2017). Total plant cover (percent cover) was visually estimated followed by a visual estimation of the absolute cover (percent cover) of each plant species within the quadrat. Additionally, the water depth was measured in the center of each quadrat and the pattern of spatial distribution was noted for each species.

Vouchers of plant specimens that were unidentifiable in the field were collected, pressed, and rapidly dried using a forced-air space heater (Blanco et al., 2006). Dichotomous keys by Voss & Reznicek (2012) and Chadde (2012) were then used to identify plant specimens in the lab. While most specimens were identified to the species level, this was unachievable for some specimens due to their lack of reproductive parts and they were identified only to genus.

Plant Biomass

Peak biomass was sampled in August, because peak biomass typically occurs in mid- to late-summer in temperate climates (Cronk & Fennessy, 2001). Due to constraints on sampling time and labor, only the above-ground biomass of the emergent vegetation zones was harvested from each wetland site. Peak biomass was harvested in the same emergent vegetation zones that were sampled for plant richness and diversity, because they were previously determined to be representative of the dominant emergent plant community for each wetland. In each of the twelve wetlands' emergent vegetation zones, all of the above-ground biomass was harvested in three square-shaped, 0.5 m^2 quadrats. Again, quadrats were placed at least one meter from the vegetation zone boundaries, and they were placed two meters apart from other quadrats to avoid trampling vegetation that was to be harvested (Uzarski et al., 2017). Harvested plant material was placed in labeled paper bags and oven-dried for one week at 65° C, at which time the samples were at a stable weight. The plant material for each quadrat was then weighed, and an average biomass (g/m^2) was determined for the emergent vegetation zone of each wetland site.

Statistical Analyses

Relative cover (derived from the total quadrat cover (percent cover) and the absolute cover (percent cover) of each plant species) was used as the relative abundance value for each species, and it was used to calculate Shannon's diversity (H') for each of the wetland sites (Shannon & Weaver, 1949). Additionally, a Floristic Quality Assessment Index (FQAI) score was calculated (Equation 1) for each of the wetland sites in order to objectively quantify the quality of the sites' plant communities (Andreas, et al., 2004).

- Equation 1: $I = \sum(CC_i)/\sqrt{N_{native}}$
  - I = FQAI score
  - CC_i = coefficient of conservatism of plant species i
  - N_{native} = total number of native species present in the wetland site

ANOVAs were used to compare the species richness, diversity, FQAI scores, and peak biomass between the restored and unrestored wetlands. Linear mixed effects models were used to compare water depth, total phosphorus concentration, and total nitrogen concentration between the restored and unrestored wetlands. All statistical analyses were performed in R (2013).

Results

Water Levels and Nutrient Concentrations

Water depth was similar between restored and unrestored wetlands ($R^2 = 0.149$, $p = 0.147$) (Table 1; Figure 2). Mean water depth was 71.86 cm (SE = ± 2.23) and 52.07 cm (SE = ±3.46) in restored and unrestored wetlands, respectively.

Total phosphorus was similar between restored and unrestored wetlands ($R^2 = 0.43$, $p = 0.47$) (Table 2). In 2016, total phosphorus was significantly higher than in 2017 (p = 1.04E-12) (Figure 3). The interaction term was not significant ($p = 0.89$). In 2016, mean total phosphorus was 0.11 mg/L (SE = ± 0.01) and 0.12

mg/L (SE = ± 0.01) in restored and unrestored wetlands, respectively. In 2017, mean total phosphorus was 0.04 mg/L (SE = ± 0.01) and 0.05 mg/L (SE = ± 0.005) in restored and unrestored wetlands respectively. Total phosphorus was square root transformed.

Total nitrogen was significantly lower in restored wetlands than in unrestored wetlands (R^2 = 0.17, p = 1.62e-04) (Table 3). Total nitrogen was significantly higher in 2017 than 2016 (p = 1.43E-04). The interaction term was significant, indicating that total nitrogen in restored wetlands was higher in 2017 than 2016 (p = 7.45e-04) (Figure 4). In 2016, mean total nitrogen was 0.58 mg/L (SE = ± 0.06) and 1.36 mg/L (SE = ± 0.13) in restored and unrestored wetlands, respectively. In 2017, mean total nitrogen was 0.99 mg/L (SE = ± 0.15) and 1.20 mg/L (SE = ± 0.1) in restored and unrestored wetlands respectively. Total nitrogen was log transformed.

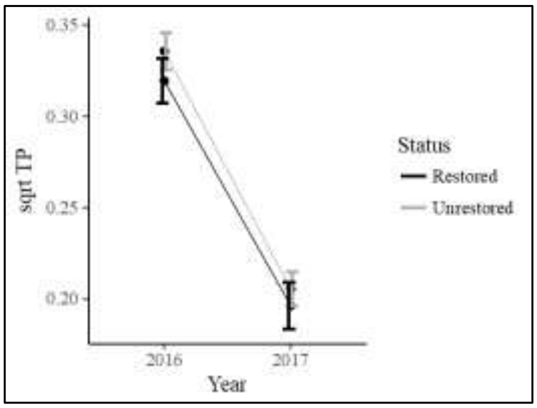

Figure 3: *Square root transformed total phosphorus in restored and unrestored wetlands in 2016 and 2017. Standard error bars also shown.*

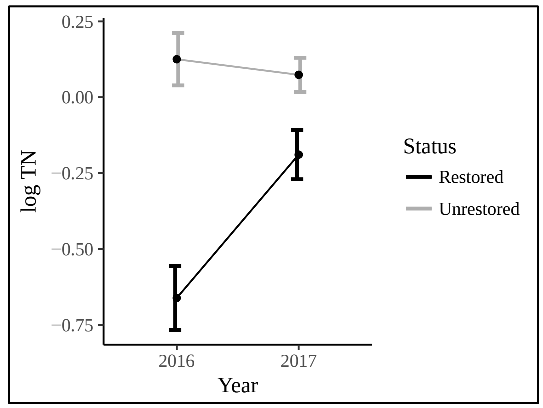

Figure 3: *Log transformed total nitrogen in restored and unrestored wetlands in 2016 and 2017. Standard error bars also shown.*

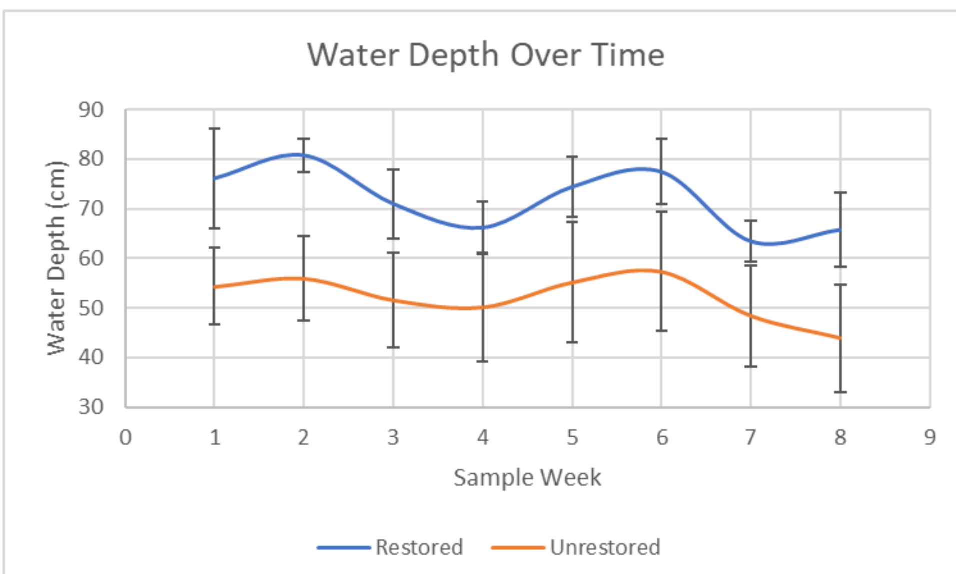

Figure 2: *A comparison of how the mean water depth (cm) changed over time between restored and unrestored wetlands. Standard error bars are also shown.*

	Estimate	Standard Error	Degrees of Freedom	t-value	p-value
(Intercept)	71.89	9.71	10.46	7.404	51.79e-05
StatusUnrestored	-19.82	12.64	10.27	-1.567	0.147

Table 1: Results of linear mixed effects model comparing water levels in restored and unrestored wetlands in 2017. Mean water level (cm) was the dependent variable. Wetland status (restored or unrestored) was a mixed effect. Site was a random effect. R^2marginal = 0.149.

	Estimate	Standard Error	Degrees of Freedom	t-value	p-value
(Intercept)	0.34	0.02	20.35	16.27	3.95e-13
StatusUnrestored	-0.02	0.02	31.88	-0.73	0.47
Year2017	-0.13	0.02	155.13	-7.76	1.04e-12
StatusUnrestored:Year 2017	0.003	0.02	153.95	0.14	0.89

Table 2: Results of linear mixed effects model comparing total phosphorus in restored and unrestored wetlands in 2016 and 2017. Square root transformed total phosphorus was the dependent variable. Wetland status (restored or unrestored), year (2016 or 2017), and the interaction term status*year were fixed effects. Site was a random effect. R^2marginal = 0.43.

	Estimate	Standard Error	Degrees of Freedom	t-value	p-value
(Intercept)	-0.61	0.14	25.83	-4.27	0.0002
StatusUnrestored	0.71	0.17	37.27	4.20	0.0002
Year2017	0.47	0.12	156.98	3.90	0.0001
StatusUnrestored:Year 2017	-0.53	0.15	156.43	-3.44	0.0007

Table 3: Results of linear mixed effects model comparing total nitrogen in restored and unrestored wetlands in 2016 and 2017. Log transformed total nitrogen was the dependent variable. Wetland status (restored or unrestored), year (2016 or 2017), and the interaction term status*year were fixed effects. Site was a random effect. R^2marginal = 0.17.

Plant Species Richness and Diversity

Plant species richness was significantly lower in restored wetlands (ANOVA, $F_{1,10}$ = 4.797, p = 0.0497) (Table 4). Mean richness was 9.4 (SE = ±0.103) and 13.14 (SE = ±0.937) in restored and unrestored wetlands, respectively.

Shannon's diversity (H') was similar between restored and unrestored wetlands (ANOVA, $F_{1,10}$ = 4.397, p = 0.0624) (Table 5). The mean H' value was 1.634 (SE = ±0.103) and 1.993 (SE = ±0.124) in restored and unrestored wetlands, respectively.

FQAI scores (I) were similar between restored and unrestored wetlands (ANOVA, $F_{1,10} = 3.205$, $p = 0.104$) (Table 6). The mean I score was 11.46 (SE = ±1.333) and 13.74 (SE = ±0.533) in restored and unrestored wetlands, respectively.

	Degrees of Freedom	Sum of Squares	Mean Square	F-value	p-value
Status	1	40.86	40.86	4.797	0.0497
Residuals	10	82.06	8.21		

Table 4: Results of ANOVA comparing plant species richness in restored and unrestored wetlands.

	Degrees of Freedom	Sum of Squares	Mean Square	F-value	p-value
Status	1	0.3759	0.3759	4.397	0.0624
Residuals	10	0.8549	0.0855		

Table 5: Results of ANOVA comparing Shannon's diversity (H') in restored and unrestored wetlands.

	Degrees of Freedom	Sum of Squares	Mean Square	F-value	p-value
Status	1	15.20	15.20	3.205	0.104
Residuals	10	47.43	4.743		

Table 6: Results of ANOVA comparing FQAI (I) scores in restored and unrestored wetlands.

Plant Biomass

Peak biomass of the emergent wetland plant communities was similar between restored and unrestored wetlands (ANOVA, $F_{1,10} = 0.423$, $p = 0.525$) (Table 7). The mean peak biomass was 1599.846 g/m^2 (SE = ±214.80) and 1843.907 g/m^2 (SE = ±270.99) for restored and unrestored wetlands, respectively.

	Degrees of Freedom	Sum of Squares	Mean Square	F-value	p-value
Status	1	173,734	173,734	0.434	0.525
Residuals	10	4,007,118	400,712		

Table 7: Results of ANOVA comparing peak biomass in restored and unrestored wetlands.

Discussion/Conclusions

While no significant difference was found in mean water depth between restored and unrestored wetlands, overall higher water levels and more water level fluctuation was observed in the mean water depth in the restored wetlands than the unrestored wetlands over the course of the 2017 sampling period (Figure 2). Presumably, the higher water levels and greater fluctuation in the restored wetlands is driven by the natural fluctuation in Crane Creek via their restored hydrological connection to Crane Creek and its tributary to Lake Eire.

Plant taxonomic richness was found to be significantly lower in restored wetlands than in unrestored wetlands, and, while not found to be statistically significant, overall lower levels of diversity (H') were also observed in the restored wetlands. Additionally, while not found to be statistically significant, mean FQAI scores (I) were found to be lower in restored wetlands than in unrestored wetlands. Lower FAQI scores are associated with a higher proportion of generalist species. These species are assigned a lower coefficient of conservatism based on their wider range of habitat preferences and ecological tolerances. In contrast, higher FQAI scores are associated with a higher proportion specialist plant species, which are assigned a higher coefficient of conservatism based on their narrower range of habitat preferences and ecological tolerances (Andreas et al, 2004). Also, while not found to be significantly different, peak biomass of the emergent wetland plant communities was observed to be lower in restored wetlands than in unrestored wetlands. Lower taxonomic richness, lower observable measures of community diversity and quality, and lower observable measures of productivity in restored wetlands than in unrestored wetlands may indicate that hydrological reconnection to the Crane Creek watershed could potentially have a negative impact on the restored plant communities' ability to mitigate the influx of nitrogen into the restored wetlands by altering previously stable ecological conditions.

While small and insignificant, there may differences in the composition and function of wetland plant communities between restored and unrestored wetlands other than taxonomic richness, which could in part be driven by more variation in hydrological conditions. There may not be significant short-term tradeoffs in wetland plant communities (other than a reduction in taxonomic richness) associated with removing dikes to reconnect Lake Erie coastal wetlands to the watershed to restore ecosystem functions. However, the longer-term monitoring of plant community composition and function will need to be carried out to identify possible trends in lower plant diversity and productivity in wetlands after the removal of dikes, which could mean potential for long-term tradeoffs in wetland plant community diversity and functioning associated with hydrologically reconnecting coastal Lake Erie wetlands to the watershed.

Site	Status	(H’)	(D)	Taxonomic Richness	FQAI_I Scores	FQAI_I' Scores	Mean Site Biomass (g/m2)
MS8A	Restored	1.877491801	0.808182995	13	15	12.5	1510.01
Pool 1	Restored	1.765124049	0.770545679	12	12.1	9.6	881.15
Pool 2A	Restored	1.267161258	0.677469136	5	7.5	5.8	2045.19
Pool 2B	Restored	1.606450766	0.760246914	7	9.5	7.2	2040.21
Pool 2C	Restored	1.652530537	0.774103704	10	13.2	11.1	1522.67
MS3	Unrestored	2.317526047	0.874466667	15	13.9	11.4	2731.81
MS4	Unrestored	2.225530732	0.86746985	14	15.7	12.6	2467.29
MS5	Unrestored	1.712892336	0.796983456	10	13	9.2	1913.29
MS7	Unrestored	1.604603221	0.722415167	10	13.2	11.1	1790.06
MS8B	Unrestored	1.664042772	0.720887964	12	12.1	9.2	868.64
Pool 3	Unrestored	2.054961959	0.80311358	15	15.6	13.9	2199.15
Pool 9E	Unrestored	2.369825773	0.882331667	16	12.7	10.5	937.11

Table 8: A summary of key characteristics for each wetland site. Includes the Shannon’s diversity value (H’), the Simpson’s diversity value (D), the taxonomic richness, the FQAI scores (I and I’), and the mean biomass (g/m^2) for each site.

References

Andreas, B.K., Mack, J.J., and McCormac, J.S. 2004. Floristic Quality Assessment Index (FQAI) for vascular plants and mosses for the State of Ohio. Ohio Environmental Protection Agency, Division of Surface Water, Wetland Ecology Group, Columbus, Ohio. 219 p.

Barry, M.J., Bowers, R., & de SZALAY, F.A. (2004). Effects of Hydrology, Herbivory and Sediment Disturbance on Plant Recruitment in a Lake Erie Coastal Wetland. The American Midland Naturalist 151:217–232. doi: 10.1674/0003-0031(2004)151[0217:EOHHAS]2.0.CO;2

Blanco, M.A., Whitten, W.M., Penneys, D.S., Williams, N.H., Neubig, K.M., and Endara, L. (2006). A simple and safe method for rapid drying of plant specimens using forced-air space heaters. Selbyana 27(1): 83-87.

Bloczynski, J.A., Bogart, W.T., Hobbs, B.F., & Koonce, J.F. (2000). Irreversible investment in wetlands preservation: Optimal ecosystem restoration under uncertainty. Environmental Management 26:175–193. doi: 10.1007/s002670010080

Chadde, S.W. (2012). *A Great Lakes Wetland Flora* (4th Ed.). CreateSpace Independent Publishing Platform: Scotts Valley, CA.

Cronk, J.K. and Fennessy, M.S. (2001). Chapter 6: The primary productivity of wetland plants. *Wetland plants: Biology and ecology* (1st Ed.). CRC Press: Boca Raton, FL.

Herdendorf, C.E. (1987). The Ecology of the Coastal Marshes of Western Lake Erie: A Community Profile. U.S. Fish and Wildlife Service Biology Report 85:171.

Kowalski, K.P., Wiley, M.J., & Wilcox, D.A. (2014). Fish Assemblages, Connectivity, and Habitat Rehabilitation in a Diked Great Lakes Coastal Wetland Complex. Transactions of the American Fisheries Society 143:1130–1142. doi: 10.1080/00028487.2014.911207

Little, A. (2013). Chapter 5: Sampling and analyzing wetland vegetation. In J.T. Anderson and C.A. Davis (Eds.), *Wetland Techniques* (273-324). Springer Science+Business Media: Dordrecht.

Lougheed, V.L., Crosbie, B., and Chow-Fraser, P. (2001). Primary determinants of macrophyte community structure in 62 marshes across the Great Lakes basin: Latitude, land use, and water quality effects.

Mitsch, W.J., Zhang. L., & Anderson, C.J. (2005). Creating riverine wetlands: Ecological succession, nutrient retention, and pulsing effects. Ecological Engineering. pp 510–527

R Core Team (2013). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL http://www.R-project.org/

Sherman D.E., Kroll, R.W., & Engle, T.L. (1996). Flora of a Diked and an Undiked Southwestern Lake Erie Wetland. The Ohio Journal of Science 96:4–8

Shannon, C.E. & Weaver, W. (1949). The mathematical theory of communication. University of Illinois Press, Urbana

Stohlgren, T.J. (2006). *Measuring plant diversity: Lessons from the field* (1st Ed.). Oxford University Press: Oxford, UK.

Tilman, D. (1999). The ecological consequences of changes in biodiversity: a search for general principles. Ecology 80:1455–1474.

Uzarski, D.G., Brady, V.J., Cooper, M.J., Wilcox, D.A., Albert, D.A., Axler, R.P, Bostwick, P., Brown, T.N., Ciborowski, J.J.H., Danz, N.P., Gathman, J.P., Gehring, T.M., Grabas, G.P., Garwood, A., and Howe, R.W. (2017). Standardized measures of coastal wetland condition: Implementation at a Laurentian Great Lakes basin-wide scale. Wetlands 37(1): 15-32.

Voss, E.G. and Reznicek, A.A. (2012). *Field Manual of Michigan Flora* (1st Ed.). University of Michigan Regional: Ann Arbor, MI.

Woodward, G. (2009). Biodiversity, ecosystem functioning and food webs in fresh waters: Assembling the jigsaw puzzle. Freshwater Biology 54:2171–2187. doi: 10.1111/j.1365- 2427.2008.02081.x

Yachi, S. & Loreau, M. (1999). Biodiversity and ecosystem productivity in a fluctuating environment: the insurance hypothesis. Proceedings of the National Academy of Sciences of the United States of America 96:1463–8. doi: 10.1073/PNAS.96.4.1463

Zedler, J.B. & Kercher, S. (2005). Wetland Resources: Status, Trends, Ecosystem Services, and Restorability. Annual Review of Environment and Resources 30:39–74. doi: 10.1146/annurev.energy.30.050504.144248